

Title: Tree height and hydraulic traits shape growth responses across droughts in a temperate broadleaf forest

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Summary

- As climate change is driving increased drought in many forested regions around the world, mechanistic understanding of factors conferring drought resistance in trees is increasingly important. The dendrochronological record provides a window through which we can understand how tree size and species' traits shape tree growth responses during droughts.
- We analyzed tree-ring records for twelve species of an oak-hickory forest - representing 97% of woody productivity - in the 25.6-ha ForestGEO plot in Virginia (USA) to determine how tree size, microenvironment characteristics, and species' traits shaped drought responses across the three strongest regional droughts over a 60-year period (1950 - 2009).
- Individual-level drought resistance decreased with tree height, which was the dominant size-related variable affecting drought response. Resistance was greater among species whose leaves lost turgor (wilted) at more negative water potentials, and whose leaves experienced less shrinkage upon dessication. However, there was substantial variation in the best predictor variables across the three drought periods.
- We conclude that hydraulic traits and tree height influence growth responses during drought, and can explain variation in the tree ring record spanning historical droughts. Thus, these traits can be useful for predicting future drought responses under climate change.

Key words: annual growth; canopy position; drought; Forest Global Earth Observatory (ForestGEO); leaf hydraulic traits; temperate broadleaf deciduous forest; tree height; tree-ring [5-8] *Five to eight key words (in alphabetical order) . Words that are in the title can, and should, be among these.*

Introduction

Forests globally play a critical role in climate regulation (Bonan, 2008), yet there remains enormous uncertainty as to how the terrestrial carbon (C) sink, which is dominated by forests, will respond to climate change (Friedlingstein et al., 2006). An important aspect of this uncertainty lies with physiological responses of trees to drought (Kennedy et al., 2019). In many forested regions around the world, the risk of severe drought is increasing (Trenberth et al., 2014; Cook et al., 2014, 2015; Mankin et al., 2019) Dai et al. 2018 **{too many}**, often despite increasing precipitation (Intergovernmental Panel on Climate Change, 2015). Droughts, exasperated by climate change, have been affecting forests worldwide (Allen et al., 2010), and are expected to continue as one of the most important drivers of forest change in the future (Anderegg et al., 2013; Adams et al., 2009; Bréda et al., 2006; Hartmann et al., 2018; Allen et al., 2015; Abrams and Nowacki, 2016; Adams et al., 2017)**{too many refs here. reduce to 2-3}**. Understanding and modeling forest responses to drought requires elucidation of how tree size, microenvironment, and species' traits jointly influence individual-level drought resistance, and the extent to which their influence is consistent across droughts. However, it has proved difficult to resolve factors affecting tree growth during drought with available forest census data, which only rarely captures extreme drought, and with tree-ring records, which rarely represent many species and size classes at a single site and are not necessarily associated with ecological data. The aim of this study was to pair tree-ring and forest plot data to test how tree size and traits shaped growth responses to historical droughts.

One fundamental question regarding forest responses to drought is what drives the observed tendency for large trees to suffer more during drought. Bennett et al. (2015) was the first to show greater growth reductions for larger trees on a global scale, and numerous subsequent studies have reinforced this finding (*e.g.*, Stovall et al. (2019); Hacket-Pain et al. (2016)), although there are exceptions (Zuleta et al., 2017). However, it has yet to be resolved which of several potential underlying mechanisms drive this pattern. First, taller trees have a greater biophysical challenge of lifting water greater distances against the effects of gravity and friction (McDowell et al., 2011; McDowell and Allen, 2015; Ryan et al., 2006; Couvreur et al., 2018), especially during drought (Zhang et al., 2009). To meet this challenge, hydraulic traits vary in relation to tree height within individuals **{REFS}**, within species **{REFS}**, and across species (Liu et al., 2019). Despite the tendency for hydraulic traits to become more conservative with height on an individual tree **{REFS}**, the need for greater hydraulic efficiency (*i.e.*, larger xylem conduits) in tall trees (Liu et al., 2019)**{REFS}** may leave them more vulnerable during drought **{REFS– Mark Olsen–mentioned by Lawren}**. Second, larger trees may have lower drought resistance because of their higher crown positions, where they are exposed to higher solar radiation, greater wind speeds, and lower humidity [Koike et al. (2001); Kunert et al. (2017) **REFS-KAT}**). Subcanopy trees tend to fare better specifically due to the benefits of a buffered environment (Pretzsch et al., 2018). Potentially counteracting the biophysical challenges faced by large trees, their larger root systems confer a potential advantage in terms of allowing greater access to water; however, it appears that this effect is usually insufficient to offset the costs of height and/or crown exposure. A final mechanism that could mediate tree size-related responses to drought is how species, and their associated hydraulic traits, are distributed with respect to size (Meakem et al., 2018; Liu et al., 2019). Within a forest community, greater drought vulnerability of larger trees could occur if the larger size classes were dominated by species less adapted to handle drought. Alternatively, larger size classes may be dominated by species that are inherently better adapted to greater biophysical challenges—as is the case in tropical moist forests of Panama, where larger size classes contain greater proportions of deciduous

species (Condit et al., 2000; Meakem et al., 2018). Understanding the mechanisms driving the greater growth reductions of larger trees during drought will require sorting out the interactive effects of height, canopy position, soil water availability, and species’ traits. {ask Lawren to check revisions to this paragraph}

Debates have also arisen regarding the traits that would influence tree growth responses to drought.

Commonly-measured traits including wood density (WD) and leaf mass per area (LMA) have been linked to drought responses in some temperate deciduous forests (Abrams, 1990; Guerfel et al., 2009; Hoffmann et al., 2011) and other forest biomes around the world (Greenwood et al., 2017). However, in other cases these traits have failed to explain drought responses (**REFS**), and the direction of response is not always consistent; for instance, higher WD has been associated with greater drought resistance at a global scale (Greenwood et al., 2017) but correlated negatively with tree performance during drought in a broadleaf deciduous forest in the southeastern United States (Hoffmann et al., 2011). Thus, their role may be due to indirect correlations with other traits within life-history strategies (Hoffmann et al., 2011). Recent work has shown a great potential for hydraulic traits to predict growth and mortality responses. Hydraulic traits including water potentials at which percent loss conductivity surpass a certain threshold ($P50$, $P80$, $P88$) and hydraulic safety margin have enabled prediction of drought performance (Anderegg et al., 2018) but are time-consuming to measure and therefore infeasible for predicting or modeling drought responses in highly diverse forests (*e.g.*, in the tropics). More rapidly measurable leaf hydraulic traits with direct linkage to plant hydraulic function are emerging with potential to explain greater variation in plant distribution and function than the more commonly-measured WD and LMA (Medeiros et al., 2019). These include leaf area shrinkage upon desiccation (PLA_{dry} ; (Scoffoni et al., 2014)) and the leaf water potential at turgor loss point (π_{tlp}), *i.e.*, the water potential at which leaf wilting occurs (Bartlett et al., 2016). The abilities of both PLA_{dry} and π_{tlp} to explain tree performance under drought remains untested.

The long time frame captured in tree-ring data enables us to address the question of whether tree size and species’ traits have similar influence across different drought events, or whether that influence is more strongly predicted by community-level responses to variable drought severity, duration, and timing based on tree size and traits. Tree growth responses vary with drought characteristics such as timing and atmospheric demand (D’Orangeville et al., 2018), but the question of how tree size and species’ traits impact growth responses across droughts still remains. While tree-ring studies provide long-term records of tree responses to multiple droughts (*e.g.*, (Lloret et al., 2011; D’Orangeville et al., 2018) **REFS**), they generally focus on species-level responses, and do not consider the roles of tree size and microenvironment. The ecological field-based studies that have shaped our understanding of the role of tree size and microenvironment in forest drought responses generally examine only a single drought and tend to focus disproportionately on the most extreme droughts with dramatic impacts (*e.g.*, [Allen et al. (2010); Bennett et al. (2015); Stovall et al. (2019); Anderegg et al. (2016)]). Thus, our knowledge of forest responses to more modest but frequent droughts - *e.g.*, those with historical return intervals on the order of one to two decades - remains limited. There is evidence that the degree to which larger trees are more impacted by drought increases with the severity of drought conditions (Bennett et al., 2015; Stovall et al., 2019), so the influence of tree size may be less in weaker—but more common—droughts. Thus, while we expect many of the mechanisms shaping drought responses to be universal, we have little understanding of how tree size and traits interact with drought characteristics, and the extent to which their influence is consistent across droughts.

To yield functional understanding of how tree size, microenvironment characteristics, and species’ traits collectively shape drought responses, we test a series of hypotheses and associated specific predictions (Table

1) based on the combination of tree-ring records from three droughts (1966, 1977, 1999), species functional and hydraulic trait measurements, and forest census data from a 25.6-ha ForestGEO plot in Virginia (USA). First, we focus on the role of tree size and its interaction with microenvironment. We test whether, consistent with most forests globally, larger-diameter trees tend to have lower drought resistance in this forest, which is in a region (eastern North America) represented by only two studies in (Bennett et al., 2015). We then test hypotheses designed to disentangle the relative importance of tree height; crown exposure; and soil water availability, which should be greater for larger trees in dry but not in perpetually wet microsites. Second, we focus on the role of species’ functional and hydraulic traits, testing the hypothesis that species’ traits—particularly leaf hydraulic traits—predict Rt. We test predictions that drought resistance is correlated with wood density—either positively (Greenwood et al., 2017) or negatively (Hoffmann et al., 2011)—and positively correlated with specific leaf area, but that hydraulic leaf traits including PLA_{dry} and π_{tlp} are better predictors. We expect that ring-porous species would have greater drought resistance than semi-ring and diffuse-porous species, as has been previously observed (Kannenberg et al., 2019; Elliott et al., 2015; Friedrichs et al., 2009).

Materials and Methods

Study site

Research was conducted at the 25.6 ha ForestGEO (Forest Global Earth Observatory) study plot at the Smithsonian Conservation Biology Institute (SCBI) in Virginia, USA (38°53’36.6”N, 78°08’43.4”W) (Bourg et al., 2013; ?). SCBI is located in the central Appalachian Mountains at the northern edge of Shenandoah National Park. Elevations range from 273-338m above sea level with a topographic relief of 65m (Bourg et al., 2013). Climate is humid temperate, with mean annual temperature of 12.7°C and precipitation of 1005 mm during our study period (1960-2009; source: CRU TS v.4.01; Harris et al. (2014)). Dominant tree taxa within this secondary forest include *Liriodendron tulipifera*, oaks (*Quercus* spp.), and hickories (*Carya* spp.).

Data collection and preparation

All analysis beyond basic data collection was performed using R version 3.5.3 (R Core Team, 2019).

Within or just outside the ForestGEO plot, we collected data on a suite of variables including tree size, microenvironment characteristics, and species traits (Table 2). The SCBI ForestGEO plot was censused in 2008, 2013, and 2018 following standard ForestGEO protocols, whereby all free-standing woody stems ≥ 1 cm diameter at breast height (DBH) were mapped, tagged, measured at DBH, and identified to species (Condit, 1998). From this census data, we used measurements of DBH from 2008 to calculate historical DBH and data for all stems ≥ 10 cm to analyze functional trait composition relative to tree height (all analyses described below). Census data, which were last updated in 2019, are available through the ForestGEO data portal (www.forestgeo.si.edu).

We analyzed tree-ring data (cambial growth increment) from 571 trees representing the twelve species contributing most to woody aboveground net primary productivity ($ANPP_{stem}$), which together comprised 97% of study plot $ANPP_{stem}$ between 2008 and 2013 (Helcoski et al., 2019) (Figure S1). Cores were collected within the ForestGEO plot at breast height (1.3m) in 2010-2011 or 2016-2017 using a 5mm increment borer. In 2010-2011, cores were collected from randomly selected live trees of species with at least 30 individuals ≥ 10 cm DBH (Bourg et al., 2013). In 2016-2017, cores were collected from all trees found dead in the annual mortality census (Gonzalez-Akre et al., 2016). Cores were sanded, measured, and

cross-dated using standard procedures, as detailed in (Helcoski et al., 2019). The resulting chronologies were published in association with Helcoski et al. (2019) (DOI: 10.5281/zenodo.2649302).

For each cored tree, we combined tree-ring records and allometric equations of bark thickness to retroactively calculate DBH for the years 1950-2009. Prior DBH was estimated using the following equation:

$$DBH_Y = DBH_{2008} - 2 * \left[\sum_{year=Y}^{2008} (r_{ring,Y} : r_{ring,2008}) - r_{bark,Y} + r_{bark,2008} \right]$$

Here, Y denotes the year of interest, r_{ring} denotes ring width derived from cores, and r_{bark} denotes bark thickness. Bark thickness was estimated from species-specific allometries based on the bark thickness data from the site (Anderson-Teixeira et al., 2015). Specifically, we used linear regression equations on log-transformed data to relate bark thickness to diameter inside bark from 2008 data (Table S1), which were then used to determine bark thickness in the retroactive calculation of DBH.

Height measurements (H , $n=1518$ trees) were taken by several researchers between 2012 to 2019, and are archived in a public GitHub repository. Measurement methods included manual (Stovall et al., 2018a; NEON, 2018), digital rangefinders (Anderson-Teixeira et al., 2015; NEON, 2018), and automatic, ground-based LiDAR (Stovall et al., 2018b). Rangefinders used either the tangent method (Impulse 200LR, TruPulse 360R) or the sine method (Nikon ForestryPro) for calculating heights. Both methods are associated with some error (Larjavaara and Muller-Landau, 2013), but in this instance there was no clear advantage of one or the other. Measurements from the National Ecological Observatory Network (NEON) were collected nearby the ForestGEO plot following standard NEON protocol, whereby vegetation of short stature was measured with a collapsible measurement rod, and taller trees with a rangefinder (NEON, 2018). Species-specific height allometries were developed (Table S2) using logarithmic regression ($\ln[H]$). For species with insufficient height data to create reliable species-specific allometries, heights were calculated from an equation developed using all height measurements.

Crown position (CP)—a categorical variable including dominant, co-dominant, intermediate, and suppressed—was recorded for all cored trees that remained standing during the growing season of 2018 following the protocol of Jennings et al. (1999). While some trees undoubtedly changed position in the 52 years between the 1966 drought and our observations in 2018, in this case the bias would be unlikely to result in false acceptance of our hypothesis (i.e., type I error unlikely; type II error possible). An analysis of crown position relative to height (Fig. 2d) and height changes since the beginning of the study period indicated that changes were fairly small relative to differences among canopy positions, although dominant and co-dominant trees were similar in height (Fig. S3). Specifically, average tree height growth was confined to $\sim 0.82\text{m}$ from 1966 to 1977, $\sim 1.45\text{m}$ from 1977 to 1999, and $\sim 1.97\text{m}$ from 1999 to 2018.

While some canopy positions undoubtedly shifted in the $\#$ years between the 1966 drought and our observations in 2018, change was relatively slow, and bias would more likely result in type x error with regards to testing our hypothesis.

Topographic wetness index (TWI) was calculated using the dynatopmodel package in R (Figure S1) (?). Originally developed by Beven and Kirkby (1979), TWI was part of a hydrological run-off model and has since been used for a number of purposes in hydrology and ecology (Sørensen et al., 2006). TWI calculation depends on an input of a digital elevation model (DEM), and from this yields a quantitative assessment

defined by how “wet” an area is, based on areas where run-off is more likely. From our observations in the plot, the calculation of TWI performed comparatively better at categorizing wet areas than the calculation of a distance matrix from a stream shapefile.

Hydraulic traits were collected at SCBI (Table 3) in August 2018. We sampled small sun-exposed branches within eight meters of the ground from three individuals of each species in and around the ForestGEO plot. Sampled branches were re-cut under water at least two nodes above the original cut and re-hydrated overnight in covered buckets (opaque plastic bags) before measurements were taken. Rehydrated leaves taken towards the apical end of the branch ($n=3$ per individual: small, medium, and large) were scanned, weighed, dried at 60°C for ≥ 48 hours, and then re-scanned and weighed. Leaf area was calculated from scanned images using the LeafArea R package (?). *LMA* was calculated as the ratio of leaf dry mass to fresh area. *PLA* was calculated as the percent loss of area between fresh and dry leaves. *WD* was calculated for $\sim 1\text{cm}$ diameter stem samples (bark and pith removed) as the ratio of dry weight to volume, which was estimated using Archimedes’ displacement. We used the rapid determination method of Bartlett et al. (2012) to estimate water potential at turgor loss point (π_{tlp}). Briefly, two 4mm diameter leaf discs were cut from each leaf, tightly wrapped in foil, submerged in liquid nitrogen, perforated 10-15 times with a dissection needle, and then measured using a vapour pressure osmometer (VAPRO 5520, Wescor, Logan, UT, USA). Osmotic potential (π_{osm}) given by the osmometer was used to estimate (π_{tlp}) using the equation $\pi_{tlp} = 0.832\pi_{osm}^{-0.631}$ (Bartlett et al., 2012). Data for hydraulic traits are available at [**create new public GitHub repo for hydraulic traits, archive in Zenodo, give DOI**].

To characterize how environmental conditions vary with height, data were obtained from the National Ecological Observation Network (NEON) tower located $<1\text{km}$ from the study area. We used data on wind speed, relative humidity, and air temperature, all measured over a vertical profile spanning from 7.2 m height to above the top of the tree canopy (31.0 or 51.8m, depending on censor), for the years 2016-2018 (?). After filtering for missing and outlier values, the data were consolidated to represent the mean values per sensor height per day. The range of these means were then aggregated at a month scale.

Identification of drought years

We identified droughts within the time period 1950-2009, defining drought (Slette et al., 2019) as events with widespread reductions in tree growth and anomalously dry peak growing season climatic conditions. Simultaneous consideration of both meteorological conditions and tree growth ensured that drought was the primary driver of observed growth declines and that our focus remained on droughts that substantially impacted the forest community. Candidate drought years were defined as those where $>25\%$ of the cored trees experienced $>30\%$ reduction in basal area increment (BAI) relative to the previous 5 years, following the drought resistance (*Rt*) metric of (Lloret et al., 2011), and identified using the pointRes package (?) in R. Specifically, *Rt* values ≤ 0.7 indicated growth reductions, whereas values ≥ 1.0 indicated increased growth. Separately, we identified the years with driest conditions during May-August, which stood out in the analysis of (Helcoski et al., 2019) as the current-year months to which annual growth was most sensitive for trees at this site. We considered two metrics of moisture deficit: NOAA Divisional Data’s Palmer Drought Severity Index (PDSI) and the difference between monthly potential evapotranspiration (PET) and precipitation (PRE). These data were obtained from the ForestGEO Climate Data Portal (<https://github.com/forestgeo/Climate>) in August 2018, with monthly PET and PRE sourced from Climatic Research Unit high-resolution gridded dataset (CRU TS v.4.01; Harris et al. (2014)). The driest years were identified through ranking mean May-August PDSI or [PET-PRE] for the time period from driest to wettest.

Statistical Analysis

For each drought period, we calculated drought resistance (Rt) as the ratio of BAI during drought to the mean BAI over the five years preceding the drought (Lloret et al., 2011). Thus, $Rt < 1$ indicates reduced growth under drought. Analyses focused on testing the predictions presented in Table 1, with either Rt or H as the response variable. The general statistical model for hypothesis testing was a mixed effects model (lme4 package from ?) with Rt as the response variable, tree nested within species as a random effect, and one or more independent variables as fixed effects. We used AICc (AICcmodavg package from ?) to assess model selection, and conditional/marginal R-squared to assess model fit.

Models were run for all drought years combined (with year as a fixed effect) and for each drought year independently. In order to determine the relative importance of the traits alone, we first tested the predictor variables independently against both height and Rt given height's substantial influence. Variables were considered useful for predicting Rt when AICc was reduced by ≥ 2 relative to the corresponding null model lacking that variable (Table 4).

Once these beset predictor variables were determined, we calculated the best overall models for predicting Rt across all traits, each individual drought year, and all years combined. We compared models with all possible combinations of candidate variables and identified the full set of models within $\Delta AICc = 2$ of the top model (that with lowest AICc). These were counted as the “top multivariate models”. When a variable appeared in all of these top models and the sign of the coefficient was consistent across models, this was counted as support for/ rejection of the associated prediction by the multivariate models. If the variable appeared in only some of the models, we considered this partial support. **see comment 27**

All data, code, and results are available through the SCBI-ForestGEO organization on GitHub (<https://github.com/SCBI-ForestGEO>: SCBI-ForestGEO-Data and McGregor_climate-sensitivity-variation repositories), with static versions corresponding to data and analyses presented here archived in Zenodo (DOIs: 10.5281/zenodo.3604993 and [TBD], respectively).

Results

Focal droughts and their characteristics {**Lawren thinks most of this (1st 3 par) belong in the methods, that par 3 might not be really useful**}

Nobby questions using “useful predictor” as a term, instead using more jargon I think In the 60-year period between 1950 and 2009, there were three droughts that met our criteria of anomalously dry climatic conditions coupled with substantial reductions in tree growth: 1966, 1977, and 1999 (Figs. 1, S2). We excluded one year (1991) meeting the growth reduction criteria (26.5% of trees experienced $>30\%$ growth reduction, mean resistance = -13.8%) because this year was not among the driest in terms of May-August [PET-PRE] or PDSI (Table S3). Rather, the severity of growth reduction may be explained in part by defoliation from gypsy moths (*Lymantria dispar* L.) from approximately 1988-1995, which strongly impacted *Quercus* spp. (Anderson-Teixeira et al., in review; Twery, 1991).

Climatically, these drought years included three of the five years between 1950 and 2009 with greatest moisture deficit (PET-PRE) during the peak growing season months of May-August (MJJA), which for this site are when annual tree growth is most sensitive (Helcoski et al., 2019). Specifically, 1966, 1977, and 1999 had mean MJJA PET-PRE of 83, 87, and 80 mm mo⁻¹, respectively. The years 1964 and 2007 also ranked in

the top five driest (PET-PRE = 84 and 82 mm mo⁻¹), but were not among the lowest in terms of PDSI and were thus not identified as candidate years for inclusion as top drought years (Table S3).

The droughts differed in intensity and prior onset (Table S3, Fig. S2). 1999 for example was the most intense (having the lowest PDSI) during May-July, while 1966 had a more intense August PDSI and overall had the second strongest drought during the growing season. The 1977 drought was the least intense throughout the growing season. All droughts increased in intensity from May through August, but differed in the previous months. The 1966 drought was preceded by two years of moderate drought during the growing season and severe to extreme drought starting the previous fall. Similarly, the 1999 drought was preceded by severe to extreme drought starting the previous fall, but before that conditions were wetter than average until the previous June. The 1977 drought was preceded by 2.5 years of near-normal conditions, and was therefore the mildest of the three droughts by any measure.

Community-level tree growth responses to these droughts were modest, with mean resistance values of 0.86, 0.84, and 0.86 for 1966, 1977, and 1999, respectively (Fig. 1b). All droughts yielded about 5% more trees experiencing negative growth ($Rt \leq 0.7$) as defined by our drought threshold, specifically, 29% in 1966, 32% in 1977, and 27% in 1999. However, some individuals exhibited increased growth: ($Rt > 1.0$): 26% in 1966, 22% in 1977, and 26% in 1999.

Tree size, microenvironment, and drought resistance

Larger-diameter trees showed greater reductions in growth during drought, although there was no significant effect during 1977 or 1999 individually (Tables 1, 4). The same held true for $\ln[H]$ as a univariate predictor (Tables 1, 4). When combined with other predictor variables, including species' traits, in our multivariate models, the top models usually included an effect of $\ln[H]$, with negative coefficient (Tables 1, 5).

Crown position varied as expected with H (dominant > co-dominant > intermediate > suppressed), but with substantial variation (Fig. 2d). Crown position was a much poorer predictor of Rt than was H (Table 4), lending little overall support to the hypothesis that trees with more exposed crowns have lower Rt (Table 1). When considered alone, CP had a significant response only in the 1966 drought, during which trees with dominant CP had the lowest Rt . When considered in conjunction with H , CP resulted as a significant predictor only for the 1999 drought, during which suppressed and then intermediate trees had the lowest Rt . Crown position was included in **almost half of the top models, with mixed results as to how Rt varied with CP (Table 5). Most commonly in these multivariate models, as in the univariate models (Table 4), the resistance of dominant trees was less than that of co-dominant trees but higher than that of suppressed trees.** Although CP was at times a useful predictor of Rt , it overall had a weak effect relative to that of H .

In the non-drought years for which we have vertical profiles in climate data (2016-2018), taller trees—or those in dominant crown positions—were generally exposed to higher evaporative demand during the peak growing season months (May-August; Fig. 2). Specifically, maximum daily wind speeds were significantly higher above the top of the canopy (40-50m) than within and below (10-30m) (Fig. 2a). Relative humidity was also somewhat lower during June-August, ranging from ~50-80 above the canopy and ~60-90% in the understory (Fig. 2b). Air temperature did not vary across the vertical profile (Fig. 2c).

Resistance was negatively correlated with $\ln[TWI]$ (Tables 4-5), negating the hypothesis that trees in moist

microsites would suffer less during drought. Nevertheless, we tested for a negative $\ln[H] * \ln[TWI]$ interaction, which could indicate that smaller trees (with smaller rooting volume) are more susceptible to drought in drier microenvironments that have a deeper water table. This hypothesis was rejected; the $\ln[H] * \ln[TWI]$ interaction was never significant and had a consistently positive coefficient (Table 4).

Species' traits and drought resistance

Hydraulic traits, including XP , PLA_{dry} , and π_{tlp} , explained variation in drought responses (Tables 1,4,5). In the *single-variable tests*, LMA and WD never significantly associated with Rt (Table 4) and were excluded from multivariable models. In contrast, XP , PLA_{dry} , and π_{tlp} all explained modest amounts of variation ($dAIC > 1.0$) in at least one drought (Table 4). PLA_{dry} was a strong predictor for the 1966 drought and all droughts combined, with consistently negative coefficients (Table 4). Similarly PLA_{dry} was consistently included, with negative coefficient, in multivariable models for the three droughts combined and for the 1966 and 1977 droughts individually (Table 5). π_{tlp} had no significant effect in the univariate models, but coefficients were consistently negative (Table 4). It was, however, included in the top multivariate model for all droughts combined and for the 1977 and 1999 droughts individually (Table 5). Xylem porosity was not significant for all droughts combined and had contrasting effects in the individual droughts: whereas ring-porous species had higher Rt than diffuse- and semi-ring- porous species in the 1966 and 1999 droughts, they had lower Rt in 1977 (Tables 4,5).

Discussion

Tree size, microenvironment, **see comment 38** and hydraulic traits shaped tree growth responses across three droughts in a temperate deciduous forest (Table 1). The greater susceptibility of larger trees to drought, similar to forests worldwide (Bennett et al., 2015), was driven primarily by their height (Liu and Muller, 1993; Stovall et al., 2019). There was a marginal additional effect of crown exposure, with the highest drought impacts felt by the most exposed and the most suppressed trees. This is consistent with observations of both the former (e.g., (Suarez et al., 2004); (Scharnweber et al., 2019)) and the latter (**REFS**). **see comment 39** There was no evidence that soil water availability increased drought resistance; in contrast, trees in wetter topographic positions experienced more growth reduction (consistent with (Zuleta et al., 2017; Stovall et al., 2019)), and larger rooting volum provided no advantage in the drier microenvironments. **see comment 40** The lower drought resistance of larger trees was not driven by a trend of more drought-sensitive species dominating the canopy. Drought sensitivity was also not consistently linked to species' LMA , WD , or xylem architecture. However, drought resistance was negatively correlated with the leaf hydraulic traits (PLA_{dry} , π_{tlp}) in the top overall model and the top models for two of the three individual droughts (Scoffoni et al., 2014; Bartlett et al., 2016; Medeiros et al., 2019). This is a novel finding in that PLA_{dry} and π_{tlp} have not previously been linked to drought growth responses. The direction of responses was mostly consistent across droughts, supporting the conclusion that they were driven by fundamental physiological mechanisms. However, the strengths of each predictor varied across droughts, indicating that specific drought characteristics interact with tree size, microenvironment, and traits to shape which individuals are most affected. These findings significantly advance our knowledge of the factors that confer vulnerability or resistance on trees during drought.

The droughts considered here were of similar severity (Fig. 1b) and fairly moderate **see comment 41**; droughts of this magnitude have occurred with an average frequency of approximately one per 10-15 years (Fig. 1a, Helcoski et al. (2019)). Therefore, we expect that most species are adapted, and individual trees

acclimatized, to survive droughts of this nature when not coupled with other disturbances. **see comment 42,43** While the majority of trees experienced reduced growth, a substantial portion had increased growth (Fig. 1b), **see comment 44** supporting the observation that these droughts did not induce extreme stress on the entire forest. It is likely for this reason, along with other non-climate factors influencing tree growth, that our best models characterize only a modest amount of variation: 11-13% for all droughts combined, and 21-26% for each individual drought (Table 5). The moderate nature of these droughts is helpful for our analysis because it considers only trees that survived all of these droughts, and we lack information on the trees that were killed prior to 2008. These are likely to be relatively modest in number, and local forest monitoring data stretching back to the late 1980s confirms that the 1999 drought did not trigger major declines in tree abundance or biomass (*Anderson-Teixeira et al., in review*). Thus, the droughts considered here are substantially weaker than others that have triggered massive tree die-off in other regions (e.g., (Allen et al., 2010)), many of which have shaped our understanding about the role of tree size (Bennett et al., 2015; Stovall et al., 2019) and-to some extent-traits underlying drought response mechanisms (Greenwood et al. (2017); Anderegg et al. (2016)). This is important because while our results are consistent with findings from more extreme, multiannual droughts, these extremes have not occurred in the Eastern United States similar to the West, and therefore we do not know exactly how forests would respond under higher mortality conditions (Clark et al., 2016; Druckenbrod et al., 2019; Berdanier and Clark, 2016). Given this consistency with our results, however, we believe this is an indication of a more intensive response.

Our analysis indicates that height—as opposed to canopy position or soil water availability—is the primary factor through which tree size mediates drought response. Taller trees face inherent biophysical challenges in lifting water a greater distance against the effects of gravity and friction (Ryan et al., 2006; McDowell and Allen, 2015; McDowell et al., 2011; Couvreur et al., 2018). Vertical gradients in stem and leaf traits—including smaller and thicker leaves (higher LMA), more negative P_{50} , and lower hydraulic conductivity at greater heights (Couvreur et al., 2018; Koike et al., 2001; McDowell et al., 2011)—make it biophysically possible for trees to become tall (Couvreur et al., 2018), yet height becomes a liability when drought incurs additional hydraulic challenges (Olson et al., 2018). Trade-offs can also occur, whereby traits conducive to efficient water transport may also lead to poor ability to recover from or re-route water around embolisms, for example in ponderosa pine (Roskilly et al., 2019). Taller trees also face different microenvironments (Fig. 2a-b), in part because they are more likely to be in dominant canopy positions (Fig. 2d). Even under non-drought conditions, evaporative demand increases with tree height while taller trees are more closely coupled to the atmosphere **see comment 45 (REFS- Jarvis 1984?;** (Bretfeld et al., 2018; Kunert et al., 2017)). Exposed canopy leaves reach higher temperatures (Smith and Nobel, 1977), particularly during drought when solar radiation tends to be higher and less water is available for evaporative cooling of the leaves. Correlation between height and canopy position (Fig. 2d) makes it challenging to disentangle the relative importance of height *per se* from microenvironment. However, significant decoupling between height and canopy position can be introduced by the configuration of neighboring trees (Fig. 2d) (Muller-Landau et al., 2006), and we show that height is a far stronger predictor of drought response than crown position for the three droughts evaluated in our study area (Tables 1,4,5).

Height and dominant canopy position had stronger negative effects in this drought than in the others (Tables 4-5), which could potentially be linked to the key reality that this was the longest and most severe multiannual drought in recent history, increasing height's propensity to be a stronger predictor of mortality (Stovall et al., 2019). see comment 49

Our analysis does have the limitation that canopy positions were re-cored in 2018 and undoubtedly changed for some trees since the 1960s, and we note that CP became an increasingly poor predictor moving back in time from 1999 to 1966 (Table 4). However, because trees would generally advance towards more dominant positions as they grow and as neighbors die, changing canopy positions would bias against the acceptance of our hypothesis. The implication is that dominant crown positions did have a marginally negative influence on Rt , which makes sense in light of the vertical environmental gradients described above and agrees with previous studies showing greater drought sensitivity in more exposed trees ((Suarez et al., 2004); (Scharnweber et al., 2019)). It is safe to assume that currently suppressed trees have always been suppressed, and their relatively low Rt (after correcting for height effects) is real, which is consistent with analyses showing that suppressed—and particularly crowded—trees can suffer disproportionately during drought (**REFS- Alan?**). **see comment 46** The observed height-sensitivity of Rt , together with the apparent lack of importance of soil water availability, agrees with the concept that physiological limitations to transpiration under drought shift from soil water availability to the plant-atmosphere interface as forests age (Bretfeld et al., 2018), such that tall trees (dominant) are the most sensitive in mature forests. Additional research comparing drought responses of young and old forest stands, along with short and tall isolated trees, would be valuable for more clearly disentangling the roles of tree height and crown exposure.

The development of tree-ring chronologies for the twelve most dominant tree species at our site (Helcoski et al., 2019; Bourg et al., 2013) gave us the sample size to compare historical drought responses across species and associated traits at a single site (see also Elliott et al., 2015). Concerted measurement of leaf hydraulic traits of emerging importance (Scoffoni et al., 2014; Bartlett et al., 2016; Medeiros et al., 2019) allowed novel insights into the role of hydraulic traits in shaping drought response. The finding that PLA_{dry} and π_{tlp} can be useful for predicting drought responses of tree growth (Tables 1,4,5) is both novel and consistent with previous studies linking these traits to habitat and drought tolerance. Previous studies have demonstrated that π_{tlp} and PLA_{dry} are physiologically meaningful traits linked to species distribution along moisture gradients (Medeiros et al., 2019; Simeone et al., 2019; Maréchaux et al., 2015) (*Rosas et al. 2019, DOI: 10.1111/nph.15684*, Fletcher et al. 2018), and our findings indicate that these traits also influence drought responses. Furthermore, the observed linkage of π_{tlp} to Rt in this forest aligns with observations in the Amazon that π_{tlp} is higher in drought-intolerant than drought-tolerant plant functional types and adds support to the idea that this trait is useful for categorizing and representing species' drought responses in models (*Powell et al. 2016, DOI: 10.1111/gcb.13731*). Because both PLA_{dry} and π_{tlp} , which can be measured relatively easily (Bartlett et al., 2012; Scoffoni et al., 2014), they hold promise for predicting drought growth responses across species. The importance of linking species' traits to drought responses increases with tree species diversity; whereas it is feasible to study drought responses for all dominant species in most boreal and temperate forests (e.g., this study), this becomes difficult to impossible for species that don't form annual rings, and for diverse tropical forests. Although progress is being made for the tropics (Schöngart et al., 2017), a full linkage hydraulic traits to drought responses would be invaluable for forecasting how little-known species and whole forests will respond to future droughts (*Powell et al. 2016, DOI: 10.1111/gcb.13731*).

As climate change drives increasing drought in many of the world's forests (Trenberth et al., 2014; Intergovernmental Panel on Climate Change, 2015), the fate of forests and their climate feedbacks will be shaped by the biophysical and physiological drivers observed here. Large trees have been disproportionately impacted in forests around the world (Bennett et al., 2015; Stovall et al., 2019), and we show, at least at this

site, that this is primarily driven by their height with some contributions from canopy position. The distinction is important because it suggests that height *per se* makes trees vulnerable, even if their crowns are somewhat protected by neighbors, whereas solitary trees or the dominant trees in young regrowth forests should be less vulnerable. This would suggest that, all else being equal, mature forests would be more vulnerable to drought than young forests with short trees; however, soil water availability may limit the young forests (Bretfeld et al., 2018), and species traits often shift as forests age. Early successional species at our site (*Liriodendron tulipifera*, *Quercus spp.*, *Fraxinus americana*) display a mix of traits conferring drought tolerance and resistance (Table 3), and further research on how hydraulic traits and drought vulnerability change over the course of succession would be valuable for addressing how drought tolerance changes as forests age (e.g. Rodríguez-Catón et al., 2015). In the meantime, the results of this study advance our knowledge of the factors conferring drought vulnerability and resistance in a mature forest, opening the door for more accurate forecasting of forest responses to future drought.

Acknowledgements see comment 51

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Author Contribution

KAT, IM, and AT designed the research. Tree-ring chronologies were developed by RH under guidance of AT and NP. Trait data was collected by IM, JZ under guidance of NK and LS. Other plot data were collected by IM, AS, EGA, and NB under guidance of EGA and WM. Data analyses were performed by IM under guidance of KAT and VH. KAT and IM interpreted the results. IM and KAT wrote the first draft of manuscript, and all authors contributed to revisions.

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